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# An impact assessment methodology for urban surface runoff quality following best practice treatment

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**Abstract** The paper develops an easy to apply desk-based semi-quantitative approach for the assessment of residual receiving water quality risks associated with urban surface runoff following its conveyance through best practice sustainable drainage systems (SUDS). The innovative procedure utilises an integrated geographical information system (GIS)-based pollution index approach based on surface area impermeability, runoff concentrations/loadings and individual SUDS treatment performance potential to evaluate the level of risk mitigation achievable by SUDS drainage infrastructure. The residual impact is assessed through comparison of the determined pollution index with regulatory receiving water quality standards and objectives. The methodology provides an original theoretically based procedure which complements the current acute risk assessment approaches being widely applied within pluvial flood risk management.

**Keywords**: impact assessment; water pollution mitigation; sustainable drainage systems (SUDS).

## 1. Introduction

Following implementation of the European Union (EU) Floods Directive, the strategic risk assessment (SRA) of urban pluvial flooding has become a legislative requirement in many countries. In the UK, the Flood and Water Management Act (2010) sets out responsibilities for producing both national and local strategic framework approaches for the identification and quantification of flood risks arising from urban pluvial flooding. The UK environmental regulatory agencies are also recommending the application of sustainable urban drainage systems (SUDS) for surface water drainage and flood control (Environment Agency (EA), 2002; Scottish Environmental Protection Agency (SEPA), 2005). However, these policy guidance documents are principally concerned with the provision of SUDS mitigation controls for attenuation and/or storage of excess surface water overland flows during wet weather conditions. This management objective has been reinforced by the requirements contained in Preliminary Flood Risk Assessments (PFRAs) and Surface Water Management Plans (SWMPs) to identify and quantify critical urban drainage areas liable to pluvial flooding (Environment Agency, 2010; Department for Environment, Food and Rural Affairs (Defra), 2010).

The issue of diffuse urban water quality has been highlighted in the risk characterisation mapping of UK river basin districts submitted under Article 5 of the Water Framework Directive (WFD). This mapping designated over 28% of UK rivers as heavily modified water bodies (HMWBs) and as being “at risk” in terms of water quality status with the majority (91%) being urban receiving waters affected through the deterioration of their physical, hydromorphic, water quality and ecological parameters. There is therefore considerable justification to seek an acceptable impact assessment methodology for the evaluation of urban non-point discharges to be used in conjunction with mitigating measures such as SUDS.

The selection and design criteria for SUDS devices in the UK are set out in guidance manuals and are essentially based on effective drainage area, site characteristics such as gradient, soil type and hydraulic infiltration rate as well as design storm event properties (Woods-Ballard et al., 2007; Ellis et al., 2008). These guideline selection criteria identify appropriate SUDS based on their volumetric capacities to control discharges for a specified storm return interval (RI) and duration e.g. 1:30 years for an event with a 10 minute duration, 1:100 years or 1:200 years for specific time durations etc.. However, it is generally recognised that although the main objective of SUDS facilities may be to address flood control, there are also water quality and ecological/amenity benefits that additionally result from their introduction into the surface water drainage network. These three prime objectives encapsulate the "formal" definition of SUDS (Campbell *et al*., 2005) as opposed to best management practices (BMPs) which also cover source control procedures such as street sweeping, gully/pipe cleansing, parking ordinances, product substitution etc..

Whilst risk assessment procedures for surface water flooding are now becoming well developed and tested in the UK (Environment Agency, 2010), there are currently no equivalent procedures for assessing the impact of stormwater runoff quality. The PFRA guidance issued by the Environment Agency for England and Wales (EA; 2010) does state that local authorities should consider the potential for urban diffuse flooding to "cause harmful consequences due to pollution", but only in relation to accident hazards, industrial/commercial pollution prevention and control (PPC) and risks to sensitive ecological sites. The SUDS drainage assessment (DA) produced by the Scottish Environment Protection Agency (SEPA; 2005) is primarily founded on general method statements which are to be included within development approval submissions with an emphasis on the calculation of rates and volumes of surface runoff for the 1:100 and 1:200 storm events. This is very similar to the advice on surface water drainage infrastructure provision contained in the Communities and Local Government (2006) PPS25 Development and Flood Risk national policy statement for England and Wales.

The Scottish DA guidance does require calculation of SUDS treatment volumes, (Vt), although no specific pollutant capture capability is specified and the recommended 4Vt specifications may well be inappropriate for many sites. According to the UK Construction Industry Research and Information Association (CIRIA) SUDS Manual (Woods-Ballard *et al*., 2007), there is a general perception that SUDS designed to the 4Vt specification will capture 90% of annual storm events including at least 65%-75% of total suspended solids (TSS) for storms up to the design event. This perception remains largely untested and there is very little generic design work that provides indicative levels of capture for other pollutants such as nutrients, hydrocarbons, metals or bacteria across the range of storm events for which SUDS are designed. There is a working assumption that infiltration-type SUDS will provide total pollutant capture but this is probably not true for many micro-pollutants which can persistently accumulate in the surface layers and act as a potential source during subsequent storm events (e.g. Birch et. al., 2005). In addition, it is also reasonable to assume that SUDS capture efficiency will decrease with time, particularly if little or no maintenance of the facility is undertaken. In this regard, no specific assessment procedure is suggested in either the EA or SEPA guidance on the quantification of absolute pollution risks arising from differing urban surfaces and sources or the relative residual risks to receiving water bodies following mitigating SUDS treatment for differing pollutant types.

To meet this need an impact assessment procedure for surface water quality which evaluates the pollutant discharge level for a stormwater SUDS drainage has been developed and is outlined in this paper. The methodology involves the identification of simple guidance on acceptable surface water drainage infrastructure provision through consideration of development site characteristics and the receiving environment. Such water quality considerations need to complement the initial application of the SUDS national manual design criteria to select appropriate drainage controls as previously mentioned. The principal drivers in surface water quality risk exposure derive from variations and intensities in urban landuse and associated impermeable surface types and activities which in turn affect the types of source pollutants flushed to the drainage network. An assessment of their relative treatability within differing SUDS facilities provides the final control driver on potential pollutant outflow loading to the receiving waterbody. It is within the context of a source-pathway-receptor modelling approach that the current impact assessment approach is based. This paper outlines a generic procedural framework for the evaluation of residual risks to receiving waters within the context of urban drainage practice, and discusses the advantages and limitations of the approach in terms of its application as a management tool for the control of diffuse urban pollution.

## 2. Risk assessment approaches

### 2.1 Joint probability analysis

Formal risk assessment approaches have proceeded by identifying potential failure mechanisms, quantifying the consequences of such failure and assessing the likelihood or probability of failure. Such a joint probability approach is well recognised and accepted (Defra, 2004) and a theoretical risk assessment framework using this approach has been developed for urban stormwater flood and pollution management (Ellis, 2010a). In this analysis the risk probabilities were considered as the product of hydraulic performance, flooding and pollution likelihood and the consequent impacts on drainage assets, receiving water quality and ecology as well as social disruption.

The mitigating SUDS controls in this risk assessment framework were considered in terms of their relative cost-performance. The methodology is essentially based on identifying the vulnerability (threats/weaknesses) of drainage system components and assessing the frequency, consequences and magnitude of likely failure with a focus on “high-level” planning aspects and stormwater issues. The resultant three-way joint risk assessment matrix can be used to generate a relative ranking of SUDS mitigating controls using a “traffic-light” display analogy of green, amber and red to identify priority threshold levels for management actions in terms of increasing concerns. This risk approach methodology has been applied within the EU 6$^{th}$ Framework Programme SWITCH project to assess surface water drainage risks in the 170 ha urban regeneration Eastside catchment in Birmingham city centre (Ellis et al., 2010).

### 2.2 GIS-based pollution index approach

It is generally acknowledged that the primary influencing factor in the quality status of urban stormwater runoff is the type, distribution and usage-intensity of directly-connected impermeable surfaces (Royal Haskoning, 2010). Surface wash-off rates by overland flow also have an effect on quality as do the mobilisation mechanisms encountered during transport over the impermeable surface to roadside gutters and gullies. In addition, further pollutant transformations will occur during residence and conveyance within the gully chamber, separate sewer pipe or open channel network. It is thus not a straightforward exercise to relate individual impermeable surface type directly to surface water quality discharging to a SUDS control facility; even relatively simple surfaces can have time-and storm-varying responses potentially undermining a lumped parameter approach.

Irrespective of the potential complexity of the quality-quantity relationships, impermeable surface type has been widely used as a guiding index to quantify the expected pollutant concentrations and loadings generated from such source landuse types during storm events. This concentration-volume approach has been widely adopted as an appropriate modelling basis (Zoppou, 2001) and has been used to successfully quantify GIS-based unit area pollution loads (UALs; kg/ha/year) for differing urban landuses in the UK (Ellis *et al*, 2006; Ellis and Revitt, 2008).

The performances of SUDS are best defined in terms of pollution capture potential for the more frequent storm events, as performance declines rapidly with increasingly extreme return periods as evidenced by the event mean concentration (EMC) data included in the US International Stormwater BMP database (2011). The majority of design guidance recommends the capture and storage of 90% of the annual average of storm events and/or the first 10 – 15mm of effective rainfall-runoff. Both these criteria relate to events which are likely to be either equal to or less than the annual average storm return interval (1:1 year RI). The UAL methodology is dependent upon defining an impermeable runoff factor ($IMP_{RF}$) for a specific urban landuse surface type which can be identified from a combination of detailed landuse maps such as the UK General Urban Landuse Data (GLUD) maps, satellite imagery, aerial photography and field inspection. This area has been well researched (Mitchell, 2001) and tested within the context of the UK Integrated Urban Drainage (IUD) pilot studies (Gill, 2008). Table 1 provides an indicative impermeability and pollution index assessment for a typical range of impermeable surface types encountered in urban areas based on data collated from a variety of UK database sources. The impermeability index values have been allocated using a range varying between 0.0 to 1.0 with the highest value representing 90%-100% impermeable cover.

2.2.1 Pollution Index (PI) assessment.

The individual pollutant indexing has been developed from consideration of the interquartile range of EMC data derived from 71 separate UK studies for a total of 205 individual storm events (Mitchell, 2001). This has been referenced against regulatory EU environmental quality standards (EQS) and Figure 1 illustrates the methodology in respect of the total suspended solids pollution index ($PI_{TSS}$). The interquartile range ($25^{th}$ and $75^{th}$ percentiles) and mean values for reported EMC data for the differing urban landuse surface types (LUST) shown in Table 1 have been used as a basis for the procedure. The pollution index (PI) is then based on a scaling of the reported event mean concentration (EMC) distribution for the given pollutant group and the likelihood that the $50^{th}$ percentile EMC values will exceed receiving waterbody environmental quality standards, specified either as a maximum allowable concentration (MAC) or annual average (AA) values.

Whilst the referencing of the $50^{th}$ percentile EMC value against the EQS 25 mg/l TSS threshold has been used as the prime derivative for the PI scaling, weighting has also been given to the interquartile spread around the mean value. One rather surprising feature of the EMC distribution shown in Figure 1 is the relatively high mean and upper quartile values reported for total suspended solids concentrations generated from open spaces. This may be partly due to a bias resulting from the small sample numbers associated with monitoring studies for this urban landuse category, but may also reflect a genuine solids yield deriving from soil erosion during higher magnitude overland flows. In addition, the compacted soils of urban open spaces can become sealed very quickly generating a substantial overland flow component and solids washoff efficiency (Ellis, 2010b). However, given the limited data availability for open space and the low sediment yields associated with the majority of storm events, this landuse category has been allocated a lower value (Table 1) than suggested by the distribution shown in Figure 1.

Table 1 Impermeability and pollution indices for different landuse types.[a]

| Landuse surface type (LUST) | Impermeability ($IMP_{RF}$) | Total suspended solids pollution index ($PI_{TSS}$) | Organic pollution index ($PI_{Org}$) | Hydrocarbon pollution index ($PI_{PAH}$) | Metals pollution index ($PI_{HM}$) |
|---|---|---|---|---|---|
| **Roofs** | | | | | |
| - Industrial/Commercial | 1.0 | 0.3 | 0.3 - 0.4 | 0.2 | 0.4 - 0.8 |
| - Residential | 0.9 | 0.4 - 0.5 | 0.6 - 0.7 | 0.1 | 0.2- 0.5 |
| **Highways** | | | | | |
| - Motorways | 0.8 - 0.9 | 0.9 | 0.7 | 0.9 | 0.8 |
| - Major arterial highways | 0.7 - 0.8 | 0.8 | 0.7 | 0.8 | 0.8 |
| - Urban distributor roads | 0.6 - 0.7 | 0.7 - 0.8 | 0.5 | 0.8 | 0.7 |
| - Residential streets | 0.4 - 0.6 | 0.4 | 0.6 | 0.6 | 0.6 |
| - Pavements | 0.5 - 0.6 | 0.4 | 0.6 | 0.3 | 0.3 |
| **Car parks/Hardstanding** | | | | | |
| - Industrial/Commercial | 0.6 - 0.8 | 0.6 – 0.7 | 0.6 - 0.7 | 0.7 | 0.4 - 0.5 |
| - Driveways (residential) | 0.5 | 0.5 | 0.6 | 0.4 | 0.3 |
| **Open Areas** | | | | | |
| - Gardens (all types) | 0.1 | 0.3 | 0.2 – 0.3 | 0 | 0.01 |
| - Parks/Golf courses | 0.2 | 0.2 - 0.3 | 0.2 | 0 | 0.02 |
| - Grassed areas (including verges; all types) | 0.1 | 0.2 - 0.3 | 0.2 - 0.3 | 0.05 | 0.05 |

[a] Pollution index values are based on reported landuse type EMC distributions and impact potential thresholds drawn from: House et al., (1993), Luker and Montague (1994), Butler and Clark (1995), Ferrier et al., (2000), Mitchell (2005) and Moy, et al., (2003).

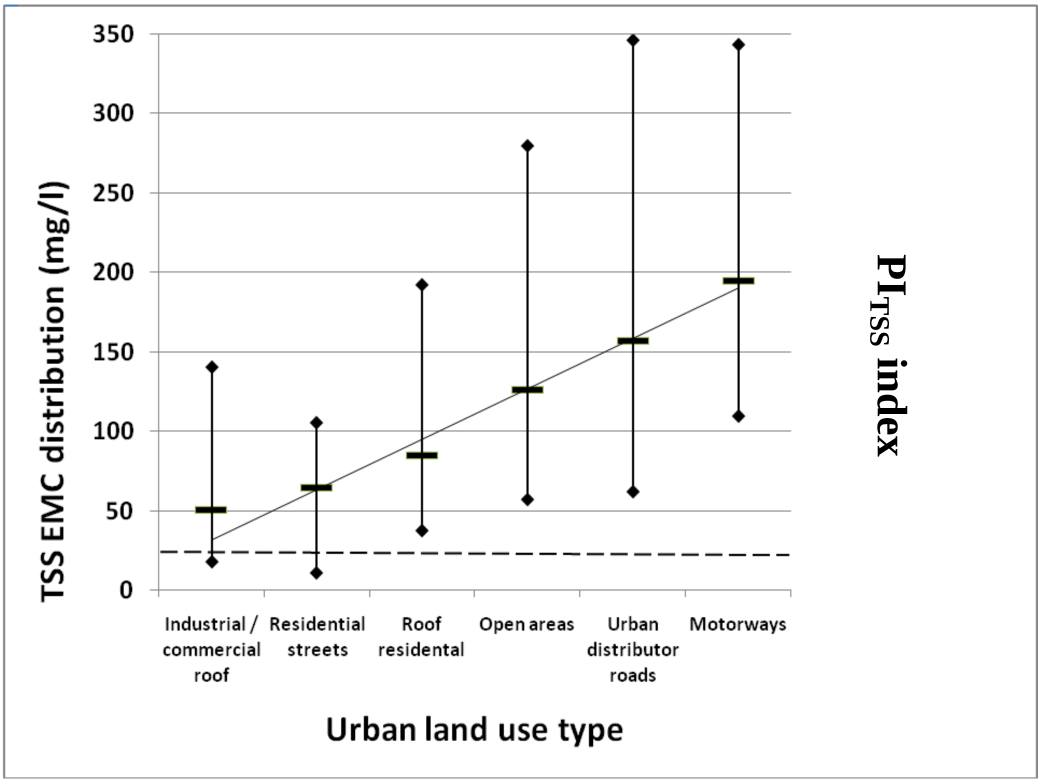


Figure 1. EMC distribution for TSS monitored in runoff from selected land use types ($75^{th}$, $50^{th}$ and $25^{th}$ percentile values identified by range bars) together with $PI_{TSS}$ index scale.

In order to evaluate the receiving water risks posed by various urban landuse EMC distributions, the 50th percentile value is deemed to be appropriate as a comparator with regulatory EQS. The $50^{th}$ percentile EMC has been traditionally used as the target descriptor for volume-concentration modelling in urban drainage studies (Zoppou, 2001). The choice of the MAC EQS threshold standard provides a direct measure of acute risk and has been used wherever possible as illustrated in Figure 1. When MAC values are not available, the annual average (AA) threshold can also be reasonably applied as a measure of potential acute impact given that the majority (>90%) of storm events are equal to or less than the annual average storm event. In addition, the AA-EQS values are significantly lower than MAC-EQS values (European Union Environmental Quality Standards, 2008), and hence their use is consistent with an initial screening approach which facilitates the assessment of a 'worst case scenario'. Other toxic thresholds that have been considered include the identification of adverse ecological standards e.g. high predicted environmental concentration (PEC):predicted no effect concentration (PNEC) ratios or the high dilution factors (e.g. <1:10) necessary to achieve a required standard. The pollutant index values shown in Table 1 include total suspended solids ($PI_{TSS}$), biodegradable organics ($PI_{Org}$), poly aromatic hydrocarbons ($PI_{PAH}$) and total heavy metals ($PI_{HM}$; represented by Cd, Cu, Ni, Pb and Zn), with lower values indicating low pollution potentials. These pollutant groupings have been selected as being typical of urban contaminants and are commonly used as general descriptors for water quality in most urban drainage studies. However, they do not necessarily represent either specific examples or the full range of pollutants which can be derived from urban surfaces.

Whilst the impermeability factors may be resilient as evidenced by various field studies (Schueler et al., 2009), the pollution index values remain to be thoroughly field tested in terms of their reliability and robustness. The previously described derivation of the $PI_{TSS}$ values has been extended to obtain the biodegradable organic pollution index ($PI_{Org}$), which is based on reported biochemical oxygen demand (BOD) concentrations (against a 30 mg/l AA EQS standard and with a MAC value ranging between 2.5 – 15 mg/l depending on water use). The EQS values for polyaromatic hydrocarbons (PAH) are expressed as total concentrations and vary between 0.002 – 0.03 µg/l depending on the specific species. The EQS values for heavy metals, represented by both MAC and AA thresholds, are generally quoted for the dissolved state (Cd, Cu, Ni and Pb) and these are therefore compared with the appropriate EMC values. In addition, metal EQS values can be dependent on water hardness (Cd, 0.08 – 0.025 µg/l as AA or 0.45 – 1.5 µg/l as MAC; Pb, 7.2 µg/l as AA; Ni, 20 µg/l; Zn, 8 – 125 µg/l and Cu, 1 - 20 µg/l) and where such EQS variations exist, a worst-case scenario has been assumed for the methodology and a comparison made of the reported EMC distributions with the lowest EQS value. The different inter-quartile ranges for the EMC values can be used as a basis for assessing the uncertainties associated with this procedure. As might be expected, the polyaromatic hydrocarbon ($PI_{PAH}$) and heavy metal ($PI_{HM}$) indices are elevated for impermeable highway surfaces in relation to average daily traffic (ADT) intensities.

2.2.2. SUDS pollution mitigation index (PMI) assessment.

A large number of approaches, guidance and manuals for selecting the most appropriate SUDS for a particular development site are available in addition to the UK national guidance manuals (Woods-Ballard *et al*., 2007). Typically they make recommendations in relation to specific site factors and represent the first-level design criteria for screening potential SUDS performance. In contrast, consideration of SUDS pollutant removal potentials has only been rarely utilised as a discriminatory criterion. Nevertheless, this performance-based criterion has become more important in terms of pollution reduction accountability contained in EU and UK river basin management planning and local/regional stormwater management plans. The Australian MUSIC model (Wong et al., 2001) was an early methodology which attempted to utilise operating processes as a basis for selecting SUDS according to their pollutant mitigation performance. This unit operating process (UOP) approach has since been fully developed to provide a full consideration of process mechanisms as a basis for identifying appropriate SUDS for treatment and control of different pollutant types (Scholes et al., 2008). The fundamental UOPs considered in the modelling approach include adsorption, settling, degradation, filtration, plant uptake, volatilisation and photolysis mechanisms.

The adopted UOP approach is similar to that followed in the US Environmental Protection Agency (EPA) SUSTAIN model (Shoemaker et al., 2009), and generates ranked preference listings of SUDS in terms of their relative performance in removing different pollutants (Scholes et al., 2008). In their paper, Scholes et al., (2008) successfully compare the theoretically derived SUDS orders of preference with field data for TSS and subject the approach to a sensitivity analysis, reporting that the results of both exercises provide an initial validation of the procedure.

Figure 2 illustrates the generated order of preference (with rank 1 being best), based on consideration of the physico-biochemical processes involved, by which different types of SUDS are able to remove total suspended solids and organic pollutants (i.e. biochemical oxygen demand, BOD) from stormwater. This hierarchy in removal potentials has been used to determine pollution mitigation indices (PMIs) for individual SUDS devices for the differing pollutant groups as shown in Table 2 for total suspended solids ($PMI_{TSS}$), polyaromatic hydrocarbons ($PMI_{PAH}$), organic pollution ($PMI_{Org}$) and heavy metals ($PMI_{HM}$).

The scaling ranges adopted are between 0.0 – 1.0 and hence qualitative i.e. a lower index value indicates a better treatment performance but the allocated scores cannot be used to indicate the magnitude of difference. Applying this scaling approach, a conventional gully-pipe drainage system would be allocated a score of 1.0 (Table 2) and thus would provide the worst mitigation option. In contrast, bio-infiltration SUDS and wet storage devices (apart from lagoons) clearly offer the best mitigation options in terms of treatment performance (Figure 2). This is associated with the ability of these systems to promote settlement/infiltration to remove TSS and additionally microbial degradation to remove organics. The extent to which these removal processes are prevalent results in the observed discrimination between the SUDS orders of preference as well as the involvement, particularly for organics, of other removal processes such as plant uptake, volatilisation and photolysis.

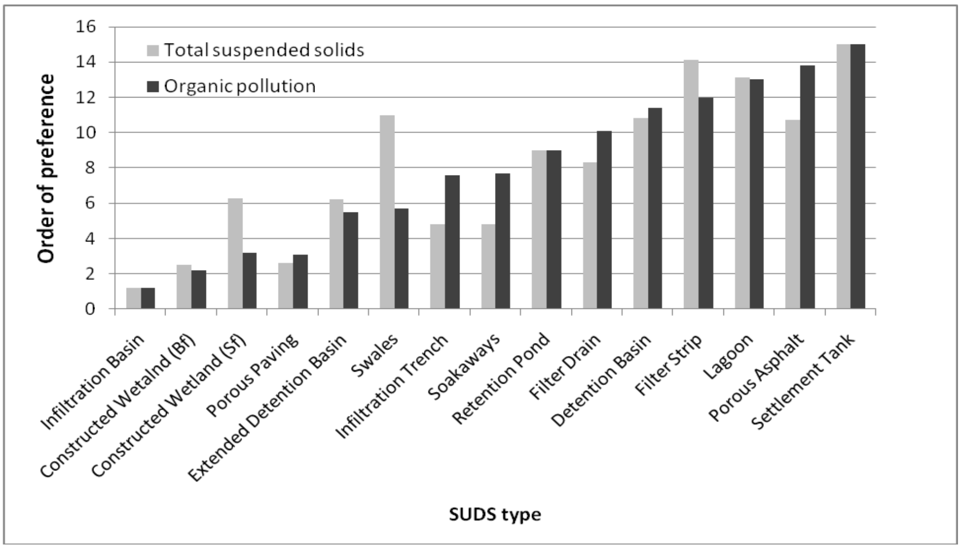


Figure 2. SUDS removal potentials for total suspended solids and biodegradable organics.

Table 2. Pollution mitigation indices for different SUDS devices and conventional pipe drainage.

| SUDS Type | Total suspended solids pollution mitigation index ($PMI_{TSS}$) | Hydrocarbon pollution mitigation index ($PMI_{PAH}$) | Organic pollution mitigation index ($PMI_{Org}$) | Heavy metal pollution mitigation index ($PMI_{HM}$) |
|---|---|---|---|---|
| Filter drains | 0.6 | 0.8 | 0.7 | 0.7 |
| Porous Asphalt | 0.7 | 0.9 | 0.9 | 0.9 |
| Porous Paving | 0.2 | 0.3 | 0.2 | 0.3 |
| Sedimentation Tank | 0.95 | 0.95 | 0.95 | 0.95 |
| Green Roof | 0.8 - 0.9 | 0.9 | 0.5 | 0.7 – 0.9 |
| Filter Strip | 0.9 | 0.8 | 0.8 | 0.7 |
| Swales | 0.7 | 0.6 | 0.4 | 0.4 |
| Soakaways | 0.3 | 0.6 | 0.5 | 0.5 |
| Infiltration Trench | 0.3 | 0.6 | 0.5 | 0.5 |
| Infiltration Basin | 0.05 | 0.05 | 0.01 | 0.05 |
| Retention Pond | 0.6 | 0.5 | 0.6 | 0.5 |
| Detention Basins | 0.7 | 0.7 | 0.8 | 0.6 |
| Extended Detention Basins | 0.4 | 0.4 | 0.4 | 0.4 |
| Lagoons | 0.9 | 0.9 | 0.9 | 0.8 |
| Constructed Wetlands | | | | |
| -Sub-surface flow | 0.2 | 0.1 | 0.1 | 0.1 |
| -Surface flow | 0.4 | 0.2 | 0.2 | 0.2 |
| Conventional gully and pipe drainage | 1.0 | 1.0 | 1.0 | 1.0 |

2.2.3 Overall site pollution index (SPI).

The three indices ($IMP_{RF}$, PI and PMI) previously described for use in the risk evaluation methodology need to be combined to define an overall site water quality assessment. This must take into consideration the flow paths followed by pollutants through each individual SUDS device proposed for a particular development site and for each specific type of landuse surface area. Taking the total suspended solids index ($PI_{TSS}$) as an example, the individual landuse area pollution index (LUPI) can therefore be defined as:

$$\text{Area LUPIi} = \text{Area LUST} \times PI_{TSS} \times [PMI_{TSS_{SUDS1}} \times PMI_{TSS_{SUDS2}} \times PMI_{TSS_{SUDS3}} \ldots\ldots\ldots n]$$

where LUST is the landuse surface type area, $PI_{TSS}$ is the suspended solids index for that surface type based on the impermeability runoff coefficient ($IMP_{RF}$) and $PMI_{TSS_{SUDS}}$ refers to the total suspended solids pollution mitigation index for each SUDS device proposed for a development site. The overall development site pollution index (SPI) will then be the sum of the GIS-area weighted LUPI (Area i....Area n) divided by the total site area.

It is recognised that there are a number of working assumptions in this theoretical procedure which can influence the risk including those which will lead to an under-estimation such as the acceptance in the UOP approach that SUDS devices will always operate at their optimum design performance and those that may produce an over-estimation such as the use of a 50 percentile EMC value as an appropriate index for measuring the impact likelihood on receiving water standards. In addition, the multiplicative procedure to derive the LUPI considers the behaviour of each SUDS on a “stand-alone” basis rather than as a component in a treatment train approach. Since the removal performances of downstream SUDS will tend to be lower due to lower pollutant exposures and capabilities, the effect of the theoretical approach will be to lower the perceived risk. It might also be argued that individual SUDS are not frequently designed to only capture and treat specific pollutant groups, but the main objective of the methodological approach is essentially to provide screening guidance on the residual water quality risk following the initial selection of SUDS facilities.

As receiving water quality standards are nearly always specified in terms of individual pollutant thresholds, it is reasonable to develop a methodology based on the consideration of separate pollutant species. As comparison of the EMC distribution is preferably with with MAC values, the procedure is essentially screening for short-term acute impact rather than any long-term chronic impacts. However, a low SPI value determined for a specific development site does not necessarily imply that low residual acute risks indicate a fully sustainable receiving water or ecological quality status over the full lifetime of the SUDS device. Persistent low level residual pollutant inputs to the receiving water environment could lead to long term prejudice of in-stream standards and this needs to be considered in any future urban extensions.

The final step in the impact assessment approach is to make a comparison of the resulting averaged SPI index, assuming a constant volume, with a recognised value or standard for receiving water quality in order to evaluate any likely risk exposure associated with the site SUDS discharge(s). One such comparison might be made against the receiving water ecosystem status, a concept similar to that embodied in the EA regulatory river ecosystem (RE) quality class objectives and in the EU HMWB ecological classification. Table 3 presents such a possible evaluation, where the higher SPI scores indicate poorer water quality and ecological status. Table 3 also identifies a number of widely used biological diversity indices including the UK Biological Monitoring Working Party (BMWP) scores and the derived Average Score Per Taxon (ASPT). The latter index is obtained by dividing the BMWP score by the number of taxa with the resulting index being independent of sample size. Full detail of these biological diversity scores and the UK RE classification as well as their relation to the EU WFD ecological status conditions can be obtained by reference to Mason (2002).

Table 3. Relationships between site pollution index (SPI) and receiving water quality

| Site Pollution Index (SPI) | Impact Level | Biological Quality | BMWP[a] | ASPT[b] | EU HMWB Ecological potential[c] | EA RE Class |
|---|---|---|---|---|---|---|
| <0.1 | Negligible | High biological diversity; several species in taxa. | >90 | >5.5 | Very good<br>Good | RE1 |
| 0.1 – 0.2 | Minimal | Small reduction in pollution tolerant taxa. | 70-90 | 5.1-5.5 | Moderate | RE2 |
| 0.2 – 0.4 | High | Many sensitive species absent; rise in pollution tolerant taxa. | 41-70 | 4.1-5.0 | Poor | RE3 |
| 0.4 – 0.7 | Substantial | Sensitive taxa scarce; some pollution tolerant species in large numbers. | 11-40 | 3.5-4.0 | Bad | RE4 |
| >0.7 | Severe | Restricted to pollution tolerant species with a few taxa dominant. | <10 | <3.5 | | RE5 |

a = Biological Monitoring Working Party classification; 0-10: heavily polluted, 11-40: polluted (ecologically impacted), 41-70: moderately impacted, 71-90: clean but slightly impacted, >90: very good; no signs of impaction.
b = Average Score Per Taxon (BMWP ÷Number of taxa which is independent of sample size.
c = European Union Heavily Modified Waterbody ecological potential.

The SPI value ranges in Table 3 have been allocated to reflect the 5 classification levels associated with EU and UK indices of ecological health. Their validity as reliable indices of SUDS surface water quality outflow status can be gauged by considering how they relate to the extremes of pollution impact. At the clean water end of the proposed scale (SPI <0.1; RE1) only PAH and heavy metals directly discharged from gardens, parks/golf courses and grassed areas are likely to preserve this pristine condition. In contrast, highly polluted discharges (SPI >0.7; RE5) would be expected for all four pollutants (TSS, organics, PAH and heavy metals) deriving from motorways and also most other types of heavily used roads. The use of a generic surface water pollution index has the advantage that it is feasible to adjust the final SPI values to take into account the impact of a specific SUDS device(s) based, for example, on a quality rating for amenity provision.

*2.2.4. Application of the SPI approach*.

The proposed procedure is applied to two examples to illustrate how it can be used to assess the urban surface runoff quality deriving from different landuses and the subsequent impact of introducing SUDS treatment. The first example relates to a small mixed landuse catchment in the regeneration area of a large city which drains into a HMWB with poor ecological status and the second example compares the impact of utilising different types of SUDS to treat the runoff from a hypothetical motorway catchment.

*2.2.4.1 Birmingham Eastside regeneration area*

Given the importance of urban HMWBs to UK receiving water quality, it is appropriate that the proposed risk evaluation methodology should be tested in a typical inner city situation, which in this example ie represented by a 4.5 ha section of the 170 ha Birmingham Eastside (UK) regeneration area. This small pilot catchment is characterised by a mix of traditional urban landuses with surface water draining to the canalised River Rea which is a typical and long standing HMWB of very poor water quality and ecology. The 4.5 ha site also represents a typical urban regeneration development area in which the municipal authorities are considering the introduction of mitigation controls to improve the receiving water quality from the current “at risk” HMWB designation.

An integrated modelling approach (SUDSLOC), based on an analysis of catchment properties including landuse type, soil type, groundwater depths, catchment area, slope etc., has identified green roof and porous paving technologies as appropriate SUDS for the mitigation of surface water flooding (Viavattene et al., 2011). The application of the SUDSLOC model predicted a runoff volume reduction averaging 22% - 25% following the introduction of SUDS control technologies to the 4.5 ha pilot catchment. Green roofs were located on all industrial premises where extensive flat surfaces were available together with half the commercial roofing area. Porous paving was assumed to be present within all car park, hardstanding and pavement areas. Table 4 indicates the PI and PMI index values for TSS and PAH for the differing urban landuse types and the resultant LUPI and SPI scores following the introduction of the SUDS controls.

Green roofs will only present a limited capability with regard to TSS and PAH removal as they only treat aerial deposition which provides a relatively small contribution to the generated total loadings of these pollutants in the overall catchment. However, it can be expected that the porous paving will divert a substantial proportion of pollutants from surface runoff discharge. The effect of the introduction of the SUDS technologies has been to equate the stormwater runoff quality, as indicated by the site SPI values, with an RE3 ecological designation compared to the before treatment situation where the site SPI value

Table 4. Pollution indices for the different landuse types in the 4.5 ha Birmingham Eastside pilot catchment

| LUST | Area (ha) | $PI_{TSS}$ (Table 1) | $PI_{PAH}$ (Table 1) | $PMI_{SUDS}$ Green Roof TSS[a] | PAH[b] | Porous Paving TSS | PAH | LUPI SPI TSS | PAH |
|---|---|---|---|---|---|---|---|---|---|
| ROOFS | | | | | | | | | |
| [c] Industrial/Commercial (treated) | 1.71 | 0.3 | 0.2 | 0.85 | 0.9 | - | - | 0.44 | 0.31 |
| Industrial/Commercial (untreated) | 0.45 | 0.3 | 0.2 | - | - | - | - | 0.14 | 0.09 |
| Residential | 0.405 | 0.45 | 0.1 | - | - | - | - | 0.18 | 0.04 |
| HIGHWAYS | | | | | | | | | |
| Distributor roads | 0.43 | 0.75 | 0.8 | - | - | - | - | 0.32 | 0.34 |
| Residential streets | 0.15 | 0.4 | 0.6 | - | - | - | - | 0.06 | 0.09 |
| [c]Pavements | 0.36 | 0.4 | 0.3 | - | - | 0.2 | 0.3 | 0.03 | 0.03 |
| CAR PARKS | | | | | | | | | |
| [c] Industrial/Commercial | 0.81 | 0.7 | 0.7 | - | - | 0.2 | 0.3 | 0.11 | 0.17 |
| Driveways | 0.05 | 0.5 | 0.4 | - | - | - | - | 0.03 | 0.02 |
| OPEN SPACE | | | | | | | | | |
| Gardens | 0.09 | 0.3 | 0 | - | - | - | - | 0.03 | 0 |
| Grassed Areas | 0.045 | 0.25 | 0.05 | - | - | - | - | 0.01 | 0.002 |
| TOTALS and LUPI SPIs | 4.5 | | | | | | | 1.35 | 1.09 |
| SITE SPI | | | | | | | | 0.3 | 0.24 |

[a]TSS = total suspended solids; [b] HC = hydrocarbons; [c] Indicates SUDS mitigation application to the surface type

for TSS (0.44) would equate to a RE4 classification. In the case of the post-treatment PAH site index, the use of SUDS reduces the SPI value to 0.24 but this remains equivalent to the RE3 classification (SPI value of 0.36) in the absence of any treatment. Therefore, even following the SUDS introduction, the discharge quality is still likely to pose considerable residual risks to the receiving water.

The receiving water of the River Rea in the inner city reaches is already a designated RE5 HMWB stream having a very poor ecological status. Therefore, whilst the low residual discharge quality of the 4.5 ha catchment imposes little additional acute risk or impact, there is clearly some minor improvement shown by individual parameters such as TSS. What is not clear from this risk analysis is whether this improvement would be substantially enhanced if SUDS controls were to be applied on a widespread scale within the entire 170 ha regeneration area. However, the reduction in overall discharge volumes resulting from the use of SUDS will certainly reduce the total pollutant loadings and this could have significance in terms of long term chronic toxic sediment accumulations.

*2.2.4.2 Hypothetical motorway case study*

To demonstrate the ability of the described methodology to differentiate between the application of different treatment options, it has been applied to a hypothetical 0.5 ha drainage area represented by a 3 lane stretch of motorway. The results are shown in Table 5 which compares the impact on discharged water quality of separately using runoff treatment by a detention basin, a surface flow constructed wetland or an infiltration basin with the no - treatment situation. For all the considered pollutants, in the absence of any treatment the quality of the motorway runoff is predicted to be of, or approaching, the lowest ecological quality as expressed by the RE classification. The ecological status is only improved modestly by using a detention basin with PAH showing the least potential to demonstrate an ecological enhancement. A surface flow constructed wetland would provide considerable improvement in the discharge quality with respect to PAH, organic pollutants and heavy metals but this SUDS is shown to be less efficient for TSS. Infiltration basins are clearly the most desirable treatment option with all investigated pollutants being removed to produce an effluent equivalent to the most desirable river ecosystem classification.

Table 5. Comparison of the ecological impacts of introducing different types of SUDS into a hypothetical motorway catchment.

| Treatment option | Pollutant | SPI | RE equivalent |
|---|---|---|---|
| No treatment | TSS | 0.9 | 5 |
| | PAH | 0.9 | 5 |
| | Org | 0.7 | 4/5 |
| | HM | 0.8 | 5 |
| Detention basin | TSS | 0.63 | 4 |
| | PAH | 0.72 | 5 |
| | Org | 0.49 | 4 |
| | HM | 0.48 | 4 |
| Surface flow constructed wetland | TSS | 0.36 | 3 |
| | PAH | 0.18 | 2 |
| | Org | 0.14 | 2 |
| | HM | 0.16 | 2 |
| Infiltration basin | TSS | 0.05 | 1 |
| | PAH | 0.05 | 1 |
| | Org | 0.07 | 1 |
| | HM | 0.08 | 1 |

*2.2.4.3 Case study perspectives*

Both case study examples indicate that SUDS implementation in urban catchments characterised by intensive landuses and having high levels of directly-connected surface water drainage will not necessarily yield rapid or high visibility improvements in receiving water quality. In these more intractable urban situations where it is considered that good ecological potential cannot be readily achieved, HMWBs can be derogated under WFD Article 4(5) and become eligible for compliance deadline extension up to 2021 to allow more time for strategic mitigation controls to be developed such as removal of in-channel contaminated sediment. In addition, the catchment examples suggest that there may be a relatively narrow range of SUDS types which are capable, in terms of either implementation or performance capabilities, of being successfully introduced into HMWB catchments. The proposed impact assessment methodology enables planners, developers and practitioners to investigate how diffuse runoff pollution from an urban development or retrofit site to a receiving water body can be minimised. The screening procedure can be applied on a targeted basis directed at SUDS type and/or the pollutant species of interest. The procedure can also be used on a 'what if' basis to evaluate the potential effect of future landuse planning changes.

The inability of current biological tools and risk procedures to fully and satisfactorily evaluate the effectiveness of mitigation measures with regard to HMWBs has been acknowledged by the EU working party reporting on HMWB classification and is also recognised by the approved description of these water bodies in terms of ecological potential criteria rather than ecological status as applied to other receiving water bodies (Royal Haskoning, 2008). The maximum endpoint for this ecological potential is that it should reflect the closest comparable non-modified surface water ecological status, although exactly how this will be achieved is contentious.

The application of ecological potential criteria to HMWBs is consonant with the so-called "alternative Prague approach" as agreed between EU member states under the Common Implementation Strategy (CIS4, 2003) of the Water Framework Directive. This alternative approach is based on the direct application of mitigation measures rather than exclusively on biological quality elements. Good ecological potential for HMWBs is then defined in terms of the biological values that might be expected from a full implementation of all possible mitigating measures as compared to the lowest possible "bad" potential having no mitigating measures applied. Such measures could include not only a checklist of generic source control measures but also water body specific mitigation measures; the use of SUDS in this sense can be applied to both approaches. Such a control-based approach avoids errors that can arise from predictive ecological modelling although there is still considerable room for translating the biological effects of mitigating measures into ecological reference values.

The proposed impact assessment methodology is fully compatible with this alternative approach and has the advantage over exclusively biological assessment approaches in that source control methods can be specifically tailored to targeted and progressive water quality and ecological improvements in order to achieve the 2021 WFD deadline for derogated HMWBs. In addition, the inclusion of water quality in the risk evaluation procedure would encourage best practice and ensure that both quantity and quality are considered in the development process for drainage infrastructure approval. A better understanding of urban surface water quality risk evaluation will support better decisions in terms of response selection and risk management strategies. Any improvement in the effectiveness of risk management approaches will serve to enhance the credibility of the risk process and ultimately lead to more reliable achievement of receiving water objectives.

## 3. Conclusions

The development and application of a structured and transparent impact assessment methodology for urban surface runoff quality complements the procedure already being applied to pluvial flood risk assessment. It is important that any such water quality impact assessment should comprise of simple, desk-based methods and tools as a means of evaluating the likely effectiveness of SUDS controls in the planning approval process and without recourse to detailed hydraulic modelling. Such impact assessment also needs to be specified in terms of receiving water quality standards and objectives in order to provide management support to the ecological concepts contained in the EU Water Framework Directive as well as to provide a potential input to any "SUDS for Adoption" standards. At its present stage of development, the generic index methodology represents a theoretical "worst-case" procedure for surface water quality risk evaluation. However, it serves to stimulate further interest in methods and tools for the assessment of residual risks associated with diffuse urban stormwater pollution control facilities for both acute and chronic conditions. This is an important objective which will certainly receive considerable future attention.

## 4. References

Birch GF, Fazell MS, Matthai C. Efficiency of an infiltration basin in removing contaminants from urban stormwater. Environ Monit Assess 2005; 101: 23–38.

Butler D and Clark P. Sediment management in urban drainage catchments. Construction Industry Research and Information Association (CIRIA); London, 1995.

Campbell N, D'Arcy B, Frost A, Novotny V Sanson A. Diffuse Pollution: An Introduction to the Problems and Solutions. International Water Association (IWA) Publishing Ltd, London, 2005.

CIS4. Identification and Designation of Heavily Modified and Artificial Water Bodies. Common Implementation Strategy. Guidance Document No.4. (CIS4). European Communities, Luxembourg. 2003. ISBN 9289451246.

Communities and Local Government. Planning Policy Statement 25: Development and Flood Risk. (PPS25). Her Majesty's Stationery Office (HMSO), London, 2006.

Department for Environment, Food and Rural Affairs (Defra). Qualitative Risk Assessment (QRA) Methodology. Defra, London, 2004.

Department for Environment, Food and Rural Affairs (Defra). Surface Water Management Plans Technical Guidance. Defra, London, 2010.

Ellis JB. Risk assessment for urban stormwater flood and pollution management. International Water Association WaterWiki. Flooding: www.iwawaterwiki.org; 2010a

Ellis JB. Managing urban runoff. In: Ferrier RC, Jenkins A, editors. Handbook of Catchment Hydrology. Wiley-Blackwell Publishing Ltd, London, 2010b, pp. 154–181.

Ellis JB, Mitchell G, Revitt DM, Scholes L. Methodologies for characterising and managing urban diffuse source pollution. In: May L, Dudley B, editors. Proceedings of the Total Maximum Daily Load Workshop, 7 April 2006, Scottish Executive, Edinburgh, Scotland. (Available from the Centre for Ecology and Hydrology, Natural Environment Research Council, Swindon).

Ellis JB and Revitt DM. Quantifying diffuse pollution sources and loads for environmental quality standards in urban catchments. Water, Air and Soil Poll 2008; 8(5/6): 577–585.

Ellis JB, Revitt DM and Scholes L. The DayWater multi-criteria comparator. In: Thevenot DR, editor. DayWater; An Adaptive Decision Support System for Urban Stormwater Management, International Water Association Publishing Ltd, London, 2008.

Ellis JB, Revitt DM and Scholes L. A risk assessment approach for priority stormwater control strategies. In: Proceedings of the 1st European International Association for Hydro-Environment Engineering and Research (IAHR) Congress, 4–6 May 2010. Heriot-Watt University, Edinburgh. Scotland; 2010.

Environment Agency. Sustainable Drainage Systems (SUDS). Policy Document EAS/0102/1/3. Environment Agency, Bristol, 2002.

Environment Agency. Preliminary Flood Risk Assessment (PFRA); Final Guidance. Report GEH0121OBTGH-E-E. Environment Agency, Bristol, 2010.

European Union Environmental Quality Standards, Directive 2008/105/EC of the European Parliament and of the Council on environmental quality standards in the field of water policy 2008 (accessed 19 7 11) at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32008L0105:EN:NOT

Ferrier RC, Jenkins A, Ellis JB, D'Arcy BJ, Dils R. Diffuse Pollution Impacts: The Environmental and Economic Impacts of Diffuse Pollution in the UK. Terence Dalton Publishers Ltd, Lavenham, 2000.

Flood and Water Management Act. Chapter 29; 2010. Retrieved from http://www.legislation.gov.uk/ukpga/2010/29/contents

Gill E. Making Space for Water: Urban Flood Risk and Integrated Drainage (HA2). Integrated Urban Drainage (IUD) Pilot Survey Report. Department of Environment, Food and Rural Affairs, London, 2008.

House MA, Ellis JB, Herricks EE, Hvitved-Jacobsen T, Seager J, Lijklema L, Aalderink H, Clifforde IT. Urban drainage: Impacts on receiving water quality. Water Sci Technol 1993; 27(12): 117–158.

Luker M and Montague K. Control of pollution from highway drainage discharges. Construction Industry Research and Information Association (CIRIA), London, 1994.

Mason, C.F. Biology of Freshwater Pollution. 4th edition. Prentice-Hall. Harlow, Essex. UK. 2002 (ISBN 0130906395).

Mitchell G. The Quality of Urban Stormwater in Britain and Europe: Database and Recommended Values for Strategic Planning Models. Technical Report, School of Geography, University of Leeds, 2001.

Mitchell G. Mapping hazards from urban diffuse pollution; A screening model to support sustainable urban drainage planning. J Environ Manage 2005; 74: 1-9.

Moy F, Crabtree RW, Simms T. Long Term Monitoring of Pollution from Highway Runoff. Environment Agency Research and Development Report No. P2-038, Water Research Centre, Swindon; 2003.

Royal Haskoning. Guidance on the Classification of Ecological Potential for Heavily Modified Water Bodies and Artificial Water Bodies. Report 9SV546 for the Environment Agency. Royal Haskoning (Environment). Westminster, London. UK. 2008.

Royal Haskoning. Non-agricultural Diffuse Pollution: Evidence Base Update. Report 9V6524 for the Environment Agency. Royal Haskoning (Environment). Haywards Heath, W Sussex, UK. 2010.

Scottish Environmental Protection Agency (SEPA). Drainage Assessment: A Guide for Scotland. Stirling, Scottish Environment Protection Agency, 2005.

Scholes L, Revitt DM, Ellis JB. A systematic approach for the comparative assessment of stormwater pollutant removal potentials. J Environ Manage 2008; 88(3): 467–478

Schueler TR, Fraley-McNeal L, Cappiella K. Is impervious cover still important: Review of recent research. J Hydrol Eng 2009; 14(4): 309–315.

Shoemaker L, Riveira J, Alvi K., Zhen JX, Paul S, Rafi T. SUSTAIN: A Framework for Placement of Best Management Practices in Urban Watersheds to Protect Water Quality. Report EPA/600/R-09/095,; US Environment Protection Agency, Office of Research and Development, Cincinnati; 2009.

Viavattene C, Ellis JB, Chlebek, J. A GIS-based integrated modelling approach for the identification and mitigation of pluvial urban flooding. 271 – 276 in Savic, D, Kapelen, Z and Butler, D (Edits): Proceedings of the Conference on Computing and Control for the Water Industry (CCWI2011). Centre for Water Systems, University of Exeter, Exeter, UK. 2011 (ISBN 0953914054).

Wong THF, Duncan HP, Fletcher TD, Jenkins GA. A unified approach to modelling urban stormwater treatment. In: Proceedings of the 2nd South Pacific Stormwater Conference, 27–29 June 2001, Auckland, New Zealand.

Woods-Ballard B, Kellagher R, Martin P, Jefferies C, Bray R, Shaffer P. The SUDS Manual. Report C697, Construction Industry Research and Information Association, London; 2007.

US International Stormwater BMP database, http://www.bmpdatabase.org (Last accessed 17 August 2011).

Zoppou, C. Review of urban storm water models. Environ Modell Softw 2001; 16(3): 195-231.

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